PERSPECTIVES IN FISH SAMPLING AND ANALYSIS TO MONITOR BIOLOGICAL INTEGRITY OF RECEIVING WATERS

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Abstract

There is a legal mandate as well as an ecological imperative to promote biological monitoring of receiving waters. There are many tools available to us, and certainly the conceptual basis of Karr's IBI model has made an important contribution to water quality assessment. I view the IBI in the context of an evolving process; the IBI is not the focal point, rather it is the community concept upon which "biotic integrity" is based that is of fundamental interest. Thus, from a national perspective it would be unwise to center on a particular, single index or phylogenetic group to monitor biological integrity; however every assessment should attempt to consider structural, functional and population characteristics (Karr 1981) which reflect water quality or habitat alteration. For several reasons, I continue to prefer to use fish to monitor water quality. However, their usage implies several constraints which I have emphasized. Precautions must be taken to maximize representative sampling. Cairns' (1977) views are appropriate: "It is evident that no single method will adequately assess biological integrity nor will any fixed array of methods be equally adequate for the diverse array of water ecosystems. The quantification of biological integrity requires a mix of assessment methods suited for a specific site and problem . . . What is needed is a protocol indicating the way in which one should determine the mix of methods that should be used to estimate and monitor threats to biological integrity."

The Legal Mandate

The environmental impact assessment (EIA) procedure and the accompanying environmental impact statement (EIS) process were legally mandated in the National Environmental Policy Act (NEPA) of 1970. NEPA was a procedural reform to institutionalize environmental considerations into the Federal planning and decision-making process (Dickson et al. 1975). The basic intent of NEPA was to require that environmental considerations be evaluated in relation to social, economic and technological factors in policy, program and project determinations. Specifically, it was required that all Federal agencies prepare a detailed EIS for actions that may affect environmental quality; environmental impacts, mitigating measures and alternatives must be considered in the EIS (Burton et al. 1983). A basic assumption of NEPA was that procedures (EIS) which generate better information will result in better decisions, however this is not guaranteed. For instance, NEPA did not prohibit authorization of projects which have adverse impact, rather it was concerned with the procedural documentation of these impacts (Fairfax & Burton 1983).

Most states have enacted similar EIS/EIA requirements in recent years. Additionally, a suite of Federal legislation was passed which strengthened NEPA in concept.

Perhaps foremost among these from an aquatic ecosystem perspective was the Federal Water Pollution Control Act of 1972 (PL92-500) which created the U.S. Environmental Protection Agency and set effluent limitations on industrial point-source discharges based on availability and economics of control technology (Hocutt 1981). The stated intention of PL92-500 was to " . . . restore and maintain the chemical, physical and biological integrity of the Nation's waters."

Other legislation impinging on the aquatic environment included the Clean Water Act, Toxic Substances Control Act and Ocean Dumping Act, among others. The Clean Water Act of 1977 amended the Federal Water Pollution Control Act of 1972 and broadened the regulations to monitor and improve water quality. The Clean Water Act defined pollution as ". . . the manmade or man-induced alteration of the chemical, physical, biological, and radiological integrity of water." Equally important to the NEPA spirit, but with a perspective of expanding public involvement, was the Freedom of Information Act of 1974 which assured public access to all public records except those falling under restricted classifications and granted citizens the right to sue those federal agencies which wrongly withhold information. In this same vein, the Federal Advisory Committee Act of 1976 and the Government in the Sunshine Act sought to increase public involvement in the decision making process, ultimately requiring that proposed Federal actions be publicly announced in the Federal Register (Fairfax & Burton 1983).

Section 304(a) of The Water Quality Act of 1987, the most recent amendment of PL92-500, has focused on the development of biological criteria and the use of instream biological data to monitor water quality. Section 304(a)(8) directs "The Administrator, after consultation with appropriate State agencies and within 2 years after the date of the enactment of The Water Quality Act of 1987, shall develop and publish information on methods for establishing and measuring water quality criteria for toxic pollutants on other basis than pollutant- by-pollutant criteria, including biological monitoring and assessment methods." In effect, the amendment emphasized the broadening of the range of criteria used to ensure compliance of standards set by the NPDES permits, and signifies a shift from pipe standards philosophy to receiving system impact.

Environmental Stress

Stress in the aquatic environment is usually viewed as man-related. However, stress may also be a natural phenomenon (Hocutt 1985); examples are (1) elevated seasonal temperatures with a corresponding decreased in saturated oxygen levels, (2) shifting substrates, and (3) fluctuations in salinity regimes. Stress can act on aquatic organisms either directly through toxic modes or indirectly through alterations in the food chain or reproductive behavior, for example. Also, stress can be viewed as being selective or non-selective in its nature. If selective, the elimination of target species with low thresholds may be observed, however this could be accompanied by increased productivity of surviving taxa. If the stress is non-selective, species richness may not decrease although overall biomass would be expected to decline.

Stress levels are usually determined by the intensity (i.e., concentration), nature (e.g., the half-life or bio-degradableness of a pollutant), mode (e.g., temperature, pH, heavy metal, pesticide, etc.), duration and rate of exposure of the organism or community to the stress. From a biological perspective, stress will be dependent upon the species, its life stage and sex, and the presence of other flora and fauna. A low-intensity stress may result in little damage even over a prolonged period of time, however, if the stress is increased either by intensity, rate of exposure or the introduction of a synergist, the probability of ecosystem damage is increased.

It is recognized that physical, chemical, radiological and biological perturbations can have a deleterious, sometimes irreversible, impact on the structure and function of impacted systems. However, it is also recognized that the environment can be used as an extension of the water treatment facility if the assimilative capacity of the system is not exceeded (Cairns 1977). Thus, environmental assessment can be viewed by two central themes: (1) water resources management, and (2) water quality assessment in terms of stress and recovery of a damaged ecosystem. It is always preferable, however, to operate within the limitations of the former to avoid the latter.

Environmental Measurement of Biological Integrity

Historically, physicochemical parameters have been given precedence over biology in the study of stressed aquatic ecosystems. Chemical evaluation of stressed conditions allows identification of the substances involved and their concentrations. This fact is central to the National Pollution Discharge Elimination System (NPDES), and its enforcement by the USEPA. However, such measurements are ineffective in estimating the synergistic affects of multiple effluents on aquatic biota, or long-term sublethal effects. Additionally, physicochemical measurements may well miss the short-term, highly concentrated discharge critical to assessment of biological impact, or other man-induced physical alterations of the environment (Karr 1981). As such ". . . pollution is essentially a biological phenomenon in that its primary effect is on living things" (Hynes 1971). Mackenthum (1969) and Hynes (1971) outlined the history of aquatic biology and its relationship to pollution effects.

Biomonitoring for NPDES compliance requirements has centered on the use of bioassay procedures rather than biosurvey methodology. Biosurveys are reported (e.g. Roop & Hunsaker 1985) to be too expensive and time consuming to warrant consideration for rapid site specific assessments; however, these arguments are weak in comparison to the fact that aquatic communities in situ are integraters of past and present environmental conditions. As well, bioassay procedures have several restrictions in their use as a holistic approach to environmental assessment: (1) laboratory-based toxicity studies may not adequately reflect ecosystem impact of point and non-point sources of discharge; (2) multiple point sources can act antagonistically or synergistically in the ecosystem; (3) there can be a large inherent variability in the toxicity tests themselves; (4) effluents have high variability, hence mean NPDES standards may not be sufficiently protective; and (5) preferred bioassay test organisms are often chosen for their tolerance to laboratory conditions, and are not necessarily the most sensitive species or life stage.

The quantification, description and comparison of terrestrial plant communities preceded similar advances for aquatic communities. Many of the biosurvey techniques used to assess aquatic ecosystems evolved from Kolkwitz and Marsson's (1908, 1909) saprobien system and Margalef's (1951) diversity index based on information theory, and resulted from the need to assess the effects of pollution. More recently James Karr and his associates have attempted (Karr 1981; Karr and Dudley 1978, 1981) to develop an index of biological integrity (IBI) using fish communities to measure stream degradation. Karr's objectives were not all together different than those of many ecologists [e.g. Cairns and Dickson (1977); Stauffer and Hocutt (1980)], i.e., to develop a system which would have predictive value for determining the amount of stress a system could assimilate, and the potential of a system to recover once it was stressed. Indeed, Karr's work (and that of others) adds emphasis to the pioneer aquatic ecology investigations of Ruth Patrick (1949), W. Beck (1954, 1955) and John Cairns (e.g. 1974) in the United States, who stressed the importance of community assemblages in data interpretation.

The emphasis of ecologists to measure "biological integrity" has been a direct consequence of the Federal Water Pollution Control Act of 1972 (PL92-500), the stated intention (to repeat from above) was to ". . . restore and maintain the chemical, physical and biological integrity of the Nation's waters." Frey (1975) defined biological integrity as "the capability of supporting and maintaining a balanced, integrative, adaptive community of organisms having a species composition similar to that of the natural habitat of the region."

I (Hocutt 1981; Hocutt and Stauffer 1980), like Karr (1981), contend that fish communities should be given preference when assessing man-related impacts in freshwaters. The most compelling reason is that structurally and functionally diverse fish communities directly and indirectly reflect water quality conditions at a given locality in that their community stability is indicative of past and present environmental perturbations (Hocutt 1981). The value of fishes in environmental assessment of estuarine and marine systems is more limited when one takes into account the large-scale migrations of many species, however, fish continue to have great utility when their seasonality of occurrence is considered in relation to their life history aspects. Stauffer and Hocutt (1980) summarized the value of using fish data in assessment of ecological integrity, noting that (1) fishes occupy the upper trophic level in most aquatic systems, and as such, the "healthiness" of the fish community implies the "healthiness" of lower trophic levels and phyletic groups, (2) in their development from larvae to mature adults, fishes pass from the primary consumer stages to subsequently higher levels, (3) fish are relatively easy to identify, thus the use of fish data is made more readily available, and (4) more is generally known for the life histories of fishes than other phyletic groups, thus it is easier to relate structural and functional relationships in fish community assemblages.

There are, however, some restrictions to the use of fisheries data for instream biomonitoring. Karr et al. (1986) identified four problem areas in sampling stream fishes accurately for an IBI analysis: (1) Purpose of data gathering must be IBI oriented to obtain a representative sample; (2) sampling gear, water conditions and fish behavior can affect accuracy; (3) the range of habitats sampled has a major effect; and (4) atypical samples result when unrepresentative habitats (e.g., beneath bridges) are next to the sample site. Additionally, I have emphasized the qualitativeness of fish collecting (Hocutt 1981), and the fact that fish may at times be totally unsuitable for monitoring ecological integrity. For instance, fish data may not accurately reflect (1) the biological purity of the water, (2) the occurrence of tastes or odors, (3) substances physically or chemically harmful to other life forms, (4) the suitability of our water source for specific industrial requirements, or (5) the desirable use of a water body for human consumption (Brown 1978).

The "advantages" of the IBI can be debated, however it remains a fact that the single most important parameter of the conceptual design of the IBI is its reliance on the structural and functional properties of the (fish community). The advantages of the IBI are reported to be: (1) It is quantitative and provides criteria to determine what is excellent or poor; (2) It uses several attributes to reflect conditions - no single attribute can reliably indicate degradation but the IBI is correlated with degradation; (3) There is no loss of information in calculating the index value -- the metric values are available to pinpoint the ecological attributes that are being altered; and, (4) Professional judgment is applied in a systematically and ecologically sound manner - this occurs when establishing metric scoring criteria, not when interpreting the index value as with most assessment methods (Miller et al. 1988). Due to the flexibility of the IBI model to be modified, it has been adapted for regulatory use in Ohio and Illinois and is currently being considered for formal adoption at the national level as a means of monitoring water quality (Miller et al. 1988).

It must be stated, however, that professional judgment remains a key issue from the moment of study design, through the field phase and especially in data interpretation. Every professional is a product of their schooling and experience; thus, while professional judgment can be a strength, it most certainly may be a weakness - and if not a weakness then a valid contrast in opinion. For example, Leonard & Orth (1986) used a modified six-metric IBI for Appalachian streams, but Angermeier & Karr (1986) included all 12 original metrics in their interpretation of the same data.

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